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engineers & scientists

Transmittal

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BLASLAND, BOUCK & LEE, INC.
6723 Towpath Road/Box 66 Syracuse, New York 13214-0066
(315) 446-9120

To: Scott Cornelius
MDEQ-ERD
301 S. Capital Avenue
Lansing, MI 48933

Date: July 19, 1999

File: 645.24.771

Re: Kalamazoo River Ecological Risk Assessment Comments
Project #: 645.24.771

We are sending you ____x____ herewith _____ under separate cover

_____ drawings _____ letters _____ other

If material received is not as listed, please notify us at once.

Quantity	Title	Action*
1	Comments on "Final Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site Baseline Ecological Risk Assessment"	A

*Action letter code: R - reviewed N - reviewed and noted I - for your information A - for your review
S - resubmit J - rejected Y - for your approval and comment

Remarks:

Enclosed please find clean copies (i.e., not faxed copies) of the comments on CDM's Ecological Risk Assessment. Please replace your current copy with the attached.

Sincerely,

BLASLAND, BOUCK & LEE, INC.

Mark Brown/ccm

Mark P. Brown, Ph.D.
Senior Vice President

MPB/ccm

cc: Ronald D. French, Camp Dresser & McKee
Tony Gendusa, Camp Dresser & McKee

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BBL
BLASLAND, BOUCK & LEE, INC.
engineers & scientists

Transmitted Via FedEx

July 16, 1999

Scott D. Cornelius
MDEQ-ERD
301 S. Capital Avenue
Lansing, MI 48933

Re: Kalamazoo River Baseline Ecological Risk Assessment
Project #: 645.24.771

Dear Scott:

On behalf of the Kalamazoo River Study Group (KRSG), Blasland, Bouck & Lee, Inc. (BBL) is submitting comments on the document titled "Final Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site Baseline Ecological Risk Assessment," which was prepared by Camp Dresser & McKee (CDM). KRSG requests that this letter and the comments prepared by Prof. John Giesy and Dr. Kenneth D. Jenkins (attached) be included in the administrative record for the site. Based on significant deficiencies in the report as outlined in this letter and in Prof. Giesy's review, the KRSG requests that these comments be addressed and the errors and inadequacies of the report be corrected prior to public release of the document.

1. In several locations within the Ecological Risk Assessment (ERA), "the NPL study area" is described as having been defined in the Michigan Environmental Response Act 307 as including 80 miles of the Kalamazoo River (e.g., pp E1, 2-1, 3-2). This is confusing. Act 307, which no longer exists, never defined "NPL study areas," nor did it establish the Kalamazoo River site on the National Priorities List (NPL). The NPL-listed site includes only 35 miles of the Kalamazoo River from the confluence of Portage Creek to the Allegan City Dam. Furthermore, contrary to the last sentence of page 3-2 the "NPL (Superfund) site" does not include the 100-year floodplain.

2. The site conceptual model and the ERA in general lacks explicit consideration of the exposed sediments in the three former impoundments. The ERA also fails to note the clear distinction between those areas and the remaining floodplain in terms of PCB levels. Surficial soil samples from the former sediment in the former Trowbridge and Plainwell impoundments should not be used to characterize floodplain soils across the site. The use of data from these terrestrial biological sampling areas (TBSAs) 3,5, and 10, should be qualified to explain that these sampling areas were purposely located on the former sediments of the Trowbridge and Plainwell Impoundments. It is misleading to characterize other floodplain surface soils throughout the site based upon these former sediment areas. Floodplain soils data presented in the MDEQ-approved Technical Memorandum 3 Results of the Floodplain Soils Investigation (TM3), which were not included in the ERA, should be used to

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describe PCB concentrations in floodplain soils adjacent to the river where appropriate throughout the site. TM3 was not even acknowledged in the ERA.

3. Data presented in the figures do not always agree with the data presented in the tables or in the sources cited. Several discrepancies were noted between data in tables and figures in the ERA, particularly in fish data. A comparison of the data presented in the ERA with the cited sources indicate that the data presented are incorrect. All numbers used in the calculations or presented in the tables or figures should be carefully checked against their respective sources to correct these errors.

4. The use of historic sediment data rather than 1993 sediment data for Portage Creek skews the results significantly. The ERA used sediment data collected between 1972 and 1983 to characterize Portage Creek. This data appears to have been taken from the Description of Current Situation (DCS), although the source was not specifically cited. Data collected in 1993 as part of the Remedial Investigation (RI) are more recent, numerous, and consistent with the other sediment data used for other ABSAs in the ERA and are a far more appropriate representation of current sediment conditions. Use of the 1993 data would reduce the mean total PCB concentration in Portage Creek sediment from 31 mg/kg to 4.2 mg/kg and the corresponding upper 95 percent confidence limit (U95) from 47 mg/kg to 7.8 mg/kg. The apparent decline in PCB levels in Portage Creek is noteworthy from a remedial planning perspective.

5. The report states that there is evidence that mink populations within the site are declining, and suggests that the decline is attributable to the presence of PCB in the various media. The basis for these statements needs to be clarified, and if a sound scientific basis cannot be provided, these statements should be retracted. It appears that this is derived from CDM's 1997 Technical Memorandum summarizing the results of the mink and muskrat sampling (where similar statements were made), it should be noted that the data in that document fail to support any such conclusion. In fact, careful review of mink sampling field notes indicates that the mink sampling was not performed in accordance with CDM's biota sampling plan and was inconsistent in both effort and timing between locations. Population "declines" inferred from the results of that trapping effort, based on CDM's field notes, could also be affected by the localized habitat (i.e., rising and filling of water levels), trap set location, trapping methods, food availability in the immediate area, the presence of humans and domestic animals in the vicinity of the traps, and the interpretation of the varied uses of lures and baited traps and stolen (or missing) traps.

6. The upper 95 percent confidence limits (U95s as abbreviated in the ERA) for PCB concentrations within individual ABSAs should not be averaged across the site (as in appendix C) because this number has no statistical meaning. Merely averaging the individual U95s does not account for PCB population distributions or individual reach characteristics, and effectively eliminates all statistical value of the data as well as any interpretive value of the results. For a representative site-wide estimate of central tendency, an area- (or length-) weighted average should be used, or the U95 should be calculated based upon the entire data set as a whole. The confidence interval is highly dependent on the sample size. Calculation of the site-wide U95 for sediment PCB based upon all of the available data (rather than the average of individual ABSA U95s based on smaller sample sizes) would reduce the site U95 approximately 40%.

7. There is no scientific basis for the speculation that the plant community growing in the residuals on the surface of the A-Site is stressed due to "contamination/toxicity" (see page 4-9).

8. If this screening level risk assessment will be used to set "recommended cleanup levels" (we believe it should not be used for that purpose), descriptions of the methods and calculations used to develop the "recommended cleanup values" and other criteria should be provided and/or explained in more detail. The mathematical basis and assumptions used to calculate many of the values are not adequately provided or well defined in the text. Specifically, the progression from individual ABSAs and TBSAs to the site as a whole should be expanded to describe how the individual numbers were combined and why.

Should you have any questions or comments, please feel free to call.

Sincerely,

BLASLAND, BOUCK & LEE, INC.

Mark P. Brown, Ph.D.
Senior Vice President

MPB/ccm
U:\TLA99\46991680.WPD

Attachment

cc: Alan J. Howard, Michigan Department of Environmental Quality
Ronald D. French, Camp Dresser & McKee
Tony Gendusa, Camp Dresser & McKee
Cynthia V. Bailey, Esq., Fort James Corporation
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Kenneth D. Jenkins, Ph.D., JSA Environmental
Douglas K. Cowin, Blasland, Bouck & Lee, Inc.
Michael D. Scoville, Blasland, Bouck & Lee, Inc.

A Critique of:

Final

Baseline Ecological Risk Assessment (ERA)

Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site

Michigan Department of Environmental Quality

Environmental Response Division

June 1999

Critique Prepared by:

Giesy Ecotoxicology, Inc.

Hilltop Place, 2355 Bravender Rd.

Williamston, MI 48895

July 16, 1999

Prepared For:

Blasland Bouck and Lee

6723 Towpath Road

Syracuse, NY 13214-0066

Executive Summary

- This report presents a review of the June 1999 final report "Kalamazoo River Ecological Risk Assessment (ERA)" prepared by Camp Dresser and McKee (CDM). While this ERA is presented as a baseline ERA, it fails to adhere to numerous EPA guidelines for baseline ERAs (EPA 1997).
- Conclusions reached regarding population effects in this document are scientifically indefensible.
- Conclusions relating to the hypotheses tested are unclear, contradictory and in general poorly written. Furthermore, the analyses presented within this ERA are incapable of directly testing any of the four listed hypotheses.
- The consistent, multiplicative use of "worst case scenario assumptions" results in highly exaggerated exposure predictions. It is appropriate to bias parameter selection to protectively compensate for uncertainty in the absence of reliable site specific data in a screening-level ERA. However, the breadth and extent of compensatory bias included within this document precludes its consideration as a baseline ERA. By definition, a baseline risk assessment is one in which both the key receptors and site specific exposure parameters have been identified and quantified respectively (EPA, 1997). As such, the necessity for bias is reduced resulting in conservative, yet predictive conclusions. Within the CDM/DEQ ERA, there were a number of prominent, easily measured parameters that were overlooked. The authors then stated that insufficient data justified the multiplicative use of "worst case scenarios". However, as in this ERA, the use of overly biased parameters such as "worst case scenario" results in a zero probability of occurrence and a large uncertainty in true predictability. The statistical combination of several worst-case scenario parameters propagates these errors in a multiplicative manner. The result is a predicted exposure level that is nonsensical because it far exceeds any maximum exposure level that is likely to occur and has massive associated uncertainty. The utilization of more appropriate values in this ERA would increase the accuracy of exposure predictions as well as reduce overall uncertainty.
- The CDM ERA and its associated uncertainties clearly illustrate the need for the quantification of additional, key, site-specific parameters.

- CDM did not thoroughly evaluate the sensitivity of their risk analyses. A basic sensitivity analysis should have been conducted regarding the selection of toxicity reference values and exposure assumptions, especially PCB content in terrestrial plants.
- The discussion on the selection of toxicity reference values contain several inaccuracies and inconsistencies. For example, all of the dietary NOAECs estimated for mink were derived by dividing the LOAEC by 10 even though CDM states on p. 4-29 that, "this ERA ... considers LOAEC/3 where NOAECs are estimated for mink with dietary exposures." Why then were the LOAECs divided by 10 rather than 3? These inaccuracies and others contribute to an overestimation of risk by at least 3 to 10-fold. Furthermore, since the LOAECs and NOAECs for mink are utilized to calculate the effects thresholds in fish as prey (0.22 mg/kg), surface water (0.00038 µg/L), and sediments (0.12 mg/kg), these errors carry through to the risk characterization and remediation discussions later in CDM's ERA. Thus, the errors committed in deriving a value for mink have significant impact on and invalidate the remaining calculations based on this value.
- On p. 4-29, CDM states that, "LOAECs are associated with adverse effects; therefore, PCB concentrations at or near the relevant LOAECs are associated with some risk." Utilizing the LOAEC values that CDM used in their risk assessment, only the mink and American robin have hazard quotients above 1.0, all of the other receptors have hazard quotients below 1.0 at the LOAEC level. Thus, given the definition of CDM, the only receptors that are associated with some risk are the mink and American robin. Given the conservative exposure assumptions and questionable toxicity reference doses, it appears that the hazard quotients for both mink and American robin are significantly less than predicted in CDM's ERA.
- Specific issues concerning the quality and handling of the data used in the assessment need to be addressed before a more comprehensive risk assessment can be conducted.

Introduction

In June 1999, Camp, Dresser, & McKee (CDM) issued a final report entitled " Baseline Ecological Risk Assessment, Allied Paper, Inc./ Portage Creek / Kalamazoo River Superfund Site. This report represents an effort at a baseline ecological risk assessment evaluating the possible environmental effects of polychlorinated biphenyls (PCBs) in the Kalamazoo River area. The report appears to be a text modified version of the previous screening level ecological risk assessment entitled "Kalamazoo River Ecological Risk Assessment (ERA)". However, no obvious additions of site-specific data have been added, such as definitively specified by EPA protocol (EPA, 1997). The major conclusion reached is that a wide range of terrestrial and aquatic biota have a high likelihood of suffering adverse effects due to exposure to PCBs under existing environmental conditions in the area surrounding the Kalamazoo River (pp. 5-18 to 5-25).

The CDM ERA was critically reviewed. Here, we present comments on the methodologies applied, assumptions made, and parameter values selected. We identify the inconsistencies and deficiencies, and indicate how the assessment would change if some of these deficiencies were corrected.

Our review indicates that this baseline ERA fails to adhere to EPA guidelines (EPA 1997). The conclusions reached in the CDM ERA are contradictory and are not supported by available data. The ERA contains several major technical and conceptual deficiencies, inappropriately conservative biases, and assumptions that are not scientifically defensible. We believe that the actual risks of PCBs to receptors are different than those stated in the CDM report due to these unsubstantiated assumptions, insufficient data and an incomplete utilization of current knowledge concerning PCB chemistry and toxicity. We believe that the conclusions presented represent a considerable overestimation of the risk associated with PCBs at this location. The collection and appropriate use of certain pieces of critical information would greatly reduce the predicted risk as well as the significant uncertainty associated with this ERA.

General Overview

The ERA conducted by CDM considered only PCBs because the National Priorities List (NPL) identifies PCBs as the major contaminant of concern. The focus of the CDM assessment on these contaminants would therefore seem to be required and appropriate for the assessment. However, available data indicates the presence of other chemical contaminants in some biota. The possibility that these other chemical stressors are having either direct effects on biota or are interacting with PCBs needs investigation. Before any risk management decisions are made, the possibility of residual risks due to these other chemical stressors should also be carefully investigated.

In addition to the potential effects of chemical stressors, lack of appropriate habitat may be a limiting factor for some of the biological receptors of interest. The impacts of physical stressors are only considered cursorily in the CDM assessment and interactions between chemical and physical stressors are not considered at all. In view of the extensively modified habitats at the site, it is essential to assess the effects of these modifications on key receptors, especially when determining assessment and remediation goals. This is particularly the case where habitat limitations currently limit the distributions and/or recovery of key receptor species (e.g. mink and bald eagle), or are likely to do so.

The term "risk" implies a probability of occurrence of an adverse effect. The overall approach of the ERA by CDM is not an evaluation of the probability of risk but rather a hazard assessment. The approach applied by CDM is a Tier I hazard assessment (HA) which applies a simple hazard quotient (HQ). In the HQ approach, point estimates are made for exposure and hazard and a simple ratio of the expected exposure to the hazard limit is calculated. The hazard represents the inherent toxicity of the compound and is generally given as a maximum acceptable toxicant concentration (MATC) or a toxicity reference value (TRV). These are concentrations of toxicant that could be allowed in the diet or tissues of a species of interest without causing observable adverse effects. The HA is a one sided test which identifies situations where no hazard is indicated. To do this, the test is structured such that there is little chance of a Type II statistical

error - determining that there is no hazard when, in fact, there is some hazard. To do this, conservative assumptions are made for exposure and hazard parameters thereby greatly increasing the chance of overestimating the potential for hazard (i.e. committing a Type I error). Hazard indices can only indicate the absence of significant risk. They cannot be used as quantitative measures of risk as is required in a baseline risk assessment. Within the decision making process it is totally inappropriate to use the hazard index approach to determine future actions except for no action or the need for further analysis. The use of hazard values for remedial action planing can lead to inappropriately biased risk-benefit analysis and to inappropriate remediation strategies. Rather, due to the conservative assumptions made and values of parameters selected in a approach such as this HQ values of greater than 20 and some times greater than 100 are required before adverse population level effects could be expected in wild populations. The limitations of using a HQ approach for ERA are well known. EPA's *Guidelines for Ecological Risk Assessment* (USEPA, 1998) indicate on p. 96 that:

> *A number of limitations restrict application of the quotient method. While a quotient can be useful in answering whether risks are high or low, it may not be helpful to a risk manager who needs to make a decision requiring an incremental quantification of risks. For example, it is seldom useful to say that a risk mitigation approach will reduce a quotient value from 25 to 12, since this reduction cannot by itself be clearly interpreted in terms of effects on an assessment endpoint.*

Several inaccuracies in understanding of the environmental fate and effects of PCBs are made in the CDM ERA. For example in Section 2, page 3, the statement "Paper wastes included kaolinite clays which concentrate PCBs like other organic materials" is incorrect. Kaolinite is not an organic material nor does it concentrate PCBs. Instead, it is the most prominent naturally occurring inorganic 1:1 (tetrahedral silica to octahedral alumina) clay. More specifically, it is a stable, non-expanding, hexagonal, inorganic clay with a low cation exchange capacity and a relatively small surface area. PCBs sorption onto kaolinite is confined to limited surface adsorption onto inorganic particles with the smallest of relative surface area. PCB sorption to

organic materials on the other hand is usually more extensive and based on a completely different mechanism called partitioning. The understanding of the differences between these two mechanisms is imperative for the accurate modeling of the fate and transport of PCBs. The statement "In addition, PCBs are persistent in the environment and degradation via chemical oxidation, hydrolysis, and photolysis in soil or aquatic systems is generally insignificant" (CDM ERA, section 2, page 3) is also inconsistent with the open literature. PCBs are degraded in the environment to different extents by each of the mechanisms listed. The most extensive of these and precursor to the others is reductive dechlorination. The extensive reductive dechlorination of PCBs has been demonstrated in the sediments of numerous rivers (Bedard, 1995; Utterman, 1996) including the Kalamazoo River (Quensen 1999). This environmental weathering process can result in significant changes in the patterns of PCB congeners such that it is inappropriate to assess environmental concentrations or predict effects based on original Aroclor mixtures.

The CDM ERA relies heavily on dated analytical technology and dated literature information. The use of Aroclor mixtures to assess the environmental fate and effects of PCBs has long been recognized as being inadequate. Their discussion presents the concept of dealing with Aroclors as single components rather than the complex mixtures that they are. Discussion of the bioaccumulation and bioavailability for "Aroclors" is not appropriate. Discussion should concern the environmental fate of individual congeners or congener groups. The congener-specific approach has been shown to be superior in evaluating both the environmental fate and biological effects of this large group of compounds and is now recommended by the EPA (U.S.EPA, Region 9, 1998). While the data available for the assessment is not congener-specific, the incorporation of this knowledge would greatly assist in the interpretation of the data used. There are many texts and papers available from the early to mid 1990s that deal with the congener-specific behavior, toxicity and assessment of PCBs. It is unclear why this literature is not utilized.

The biological effects of PCBs are expressed primarily through the *Ah*-receptor pathway (Okey *et al.*, 1994). Therefore, the most appropriate method for assessing PCB risks is to convert

congener-specific data to 'Dioxin Toxic Equivalents" using appropriate Toxic Equivalency Factor values (van den Berg *et al.*,1998). This approach is not acknowledged in the CDM assessment and as a result discussions of toxic effects based on total PCBs introduce additional uncertainty into the assessment.

The CDM ERA relies heavily on the modeling of PCB concentrations in various biotic and abiotic compartments to estimate exposure to key receptors. Available information on concentrations of PCBs in biota could have been used to more directly, and accurately, estimate exposure in some key receptors. An example of the bias introduced by the modeling approach is the unrealistically great PCB concentrations estimated for terrestrial plants which result in gross overestimates of hazard to Red fox, American robin, and White-footed deer mouse (CDM ERA, Section 4, page 24-25).

Deficiencies in the modeling process are also apparent in the use and determination of BAF values derived from water and fish data. First, while CDM refers to these values as BAFs, they are actually BCFs in that they divided concentrations of PCBs in fish divided by concentration of PCBs in water (which is by definition a BCF, discussed in more detail later). Concentrations of PCBs in water were determined for samples containing particulate matter and do not indicate the truly dissolved and bioavailable concentrations of PCBs in water. As the BAF values were not calculated using dissolved phase PCB concentrations, the BAF values can not be used in subsequent modeling.

Probably the most significant deficiency of the CDM ERA is the choice of conservative and compounding parameter estimates for the exposure analysis section of the ERA. CDM, in their ERA, frequently assume the 'worst-case scenario' and apply the most conservative estimates of critical exposure parameters (i.e. use of greatest measured PCB concentration in mice, earthworms, and muskrats as the exposure concentrations). While a baseline risk assessment should be conservative, the compounding nature of these parameter choices results in unrealistic estimates of HQs. There is by definition a virtually zero probability of a chosen worst-case

scenario occurring. In the CDM ERA, the compounding effect of these conservative biases through several trophic levels results in highly inflated hazard estimates having little basis in reality. Furthermore, the concepts of error compounding and error propagation are not discussed in the ERA.

In summary, while the ERA acknowledges considerable uncertainty inherent in the analysis, the conclusions reached in this ERA are derived from assumptions and model estimates that are not supported by the data presented. The modeled data set that CDM used is insufficient to conduct a baseline risk assessment (*e.g.*, concentrations of PCBs in terrestrial plants). Furthermore, the analyses presented in this ERA cannot support the conclusions drawn and certainly can not support decisions pertaining to the remediation goals identified.

Specific Comments

Specific sections of the CDM ERA are discussed below in the context in which they were presented in the CDM ERA.

Executive Summary

- General Approach
  The EPA guidance that is being used is not stated.

- ERA Related Goals and Objectives
  Human Health is not an ERA concern and is inappropriate in this arena.

- Site Conceptual Model
  The bulleted list includes "contaminants" (plural), however, the ERA addresses only PCBs. The text should be altered to make this clear.

- Overall Risk Summary
  Statements made within this section are inconsistent and/or contradictory such as:
  "Most aquatic biota such as invertebrates and fish are unlikely to be adversely affected". This statement contradicts the rejection of the null hypothesis which states, "levels of contaminants in water, sediment and biota are not sufficient to cause adverse alterations to the structure or function of fish populations."
  "Terrestrial and semi-aquatic biota.....risks are in general considered to be low to moderate." This statement is contradictory to the following statements, "carnivorous terrestrial species represented by the fox... are unlikely to be at significant risks..." and "omnivorous terrestrial species are also unlikely to be at significant risks".

Section 3 Problem Formulation

The first phase of the ERA is the identification of chemicals and biological receptors of concern. In this phase of the ERA, CDM focuses solely on the presence of PCBs in the environment. The NPL decision indicated that PCBs are the primary contaminant of concern and the available fish data indicates that of the persistent organochlorines analyzed, PCBs are the most commonly detected compounds and are present at the greatest concentrations. However, the effects of PCBs should be considered in the context of other stressors.

Section 3.1 Stressor Identification - Chemical Stressors

The discussion of the chemical and environmental properties of PCBs given by CDM reflects the state of knowledge in the 1980's. It is now realized that the use of Aroclor mixtures is not appropriate in discussions of the environmental behavior of PCBs (US-EPA, 1998). Each Aroclor is a complex mixture of numerous PCB congeners and each congener behaves independently once released to the environment. Due to selective degradation, accumulation, sorption and metabolism, the relative concentrations of congeners change as a function of time. Thus, it is inappropriate to quantify weathered PCBs as technical mixtures. It is also inappropriate to compare total PCB concentrations in environmental samples to the results of studies utilizing unweathered technical mixtures.

To some extent, this deficiency is caused by the presentation of the data as Aroclor concentrations. The analytical procedure used does not measure 'Aroclors'. Rather, it detects a pattern of congeners and then determines the Aroclor pattern that most closely approximates that mixture. For this reason, although a wide variety of Aroclors are detected in abiotic samples in the Kalamazoo River sediment survey (KRSS), only Aroclor 1260 is reported in biota samples (CDM ERA, Table 3-1). In this situation, the 'sorting' of the congeners passing through the various steps in the food chain results in a relatively uniform congener mixture in biota and that pattern most closely resembles Aroclor 1260 but does not represent the PCBs released to the environment or the complex mixtures that occur in the environment.

The most severe ecological effects of PCBs work through a specific mechanism of action, the *Ah*-receptor mechanism. Chemicals operating through other modes of action are toxic at considerably greater exposures and so, as stated in the CDM ERA, are generally of less significance. However, chemicals that operate through the same mode of action as PCBs must be considered in the ERA if a realistic estimate of risk is to be obtained. In particular PCDDs and PCDFs operate through the same mechanism of action as PCBs and do so at concentrations much lower than PCBs. As PCDDs and PCDFs have been measured at the KRSS (BBL 1995, Addendum 2 to Technical memorandum 14) these compounds must either be included in the

assessment or data presented to demonstrate that their contribution to the overall *Ah*-receptor active toxicity is negligible. To facilitate considering the most appropriate mechanism of action a congener-specific approach to the assessment of the ecological risks posed by PCBs would give a more accurate assessment of risk.

Additional chemicals of concern that CDM should have considered in their ERA include pesticides and metals which have been detected at elevated concentrations in biota.

Physical Stressors

The discussion of physical stressors at the site is limited. In general habitat descriptions of the river are vague and not identified by reach. Inclusion of habitat descriptions for each ABSA/TBSA would greatly assist interpretation of species abundance data. The ERA discusses some of the significant habitat changes that have occurred in the area but no attempt was made in the later parts of the assessment to incorporate these effects in the final risk characterization. Given that many of the assessment endpoints are related to wildlife populations and abundance (Section 3.3), it is critical that the effects of habitat loss or alteration be incorporated into the assessment. This is particularly important in the case for mink where the absence of this species from certain reaches of the river may be attributable to lack of suitable habitat rather than chemical contamination.

Section 3.2 Ecological Receptors Potentially at Risk

This and subsequent sections of the problem formulation phase simply list all possible resources and species. This is not problem formulation. Problem formulation should use existing knowledge to predict critical areas and receptors of concern. For example, from current knowledge it is known that fish-eating birds are exposed to significant amounts of PCBs in areas where PCB-contaminated sediments are present. Therefore, fish-eating birds such as blue heron and belted kingfisher should be considered as a specific receptor class. Conversely, PCBs do not accumulate significantly in plants, therefore plants, although they may be relevant in food chain analyses, both aquatic and terrestrial can be eliminated as key receptors. Reptiles and

amphibians are not included in the CDM ERA despite being significant aquatic species. Finally, the bald eagle is mentioned only in passing. Therefore the assessment and measurement endpoints are not consistent.

Section 3.3.1 Assessment Endpoints

The assessment endpoints chosen by CDM are vague and do not relate well to either measurement endpoints or remediation goals. Five criteria have been suggested for the evaluation of potential assessment endpoints (Suter, 1992 cited in Massachusetts DEP, 1996):

- Unambiguous operational definition, which provides direction for testing and modeling;
- Accessibility to prediction and measurement, which means that the response of an endpoint can be measured or estimated reliably from measurements of related responses or component responses;
- Susceptibility to the hazardous agent, which results from the potential for exposure and responsiveness to exposure;
- Biological relevance, which is determined by its importance to a higher level of the biological hierarchy; and
- Societal relevance, which means that the endpoint is valued by the decision-maker and the public.

While the assessment endpoints stated are highly biologically and/or socially relevant, they fall short of meeting several of the above criteria. First, there is no indication as to how the 'health' of populations is defined or measured (criterion 1). Second, while it may be possible to measure population health and status, there needs to be definition of which populations are measured and what observed changes in populations mean (criteria 1 and 2). Finally, while populations may well be susceptible to PCB effects there is no rationale for linking observed population changes specifically to PCB effects rather than other stressors.

Furthermore, the assessment endpoints chosen are abundance- and diversity-based. These endpoints do not necessarily reflect the 'health' of the population. Some population health

measures such as productivity should be included. Additional problems in this section include inconsistency between receptors of interest, null hypotheses, assessment endpoints, and receptors which actually are evaluated quantitatively in the risk assessment. For example, plants and invertebrates are listed as receptors of interest but are not included in the null hypotheses and assessment endpoints. The null hypothesis for fish refers to population "structure and function", whereas the assessment endpoint refers to "preservation" of populations.

Section 3.3.2 Measurement Endpoints

The measurement endpoints are neither very specific nor do they relate directly to the assessment endpoints. The relationships between measurement and assessment endpoints should ideally enable the risk assessor to use the results of field observations, bioassays, and literature reviews to decide whether a significant risk of harm has resulted, or is likely to result, from the contaminant(s) of concern. Using a measurement endpoint to approximate an assessment endpoint introduces uncertainty; the stronger the relationship between the assessment and measurement endpoints, the less the uncertainty. Thus, the risk assessor should attempt to minimize the uncertainty about an assessment by selecting assessment endpoints, which are closely related to available measurement methods (Massachusetts DEP, 1996).

Summary of Section 3

In general, the problem formulation section has not been completed adequately. The ERA goals are only loosely defined. Through the entire section, the ERA simply presents the total picture. While this information is required to set the scene, the problem formulation phase needs to focus the assessment on critical issues and critical receptors. The assessment endpoints would be more appropriately framed as "Maintain or enhance the health status of the ... population/community" while measurement endpoints should address the measurement of population/community parameters as well as contaminant concentrations. Finally, the development of remediation goals somewhat presupposes the outcome of the assessment.

Section 4 Analysis Phase

Toxicological information was gathered and used to formulate contaminant-response relationships for key receptors. Since extrapolation among species and between laboratory and field exposures introduces additional uncertainty, the type of data used is of great importance. Again, data should be chosen which minimizes the need for extrapolation and consequent increases in uncertainty.

All the discussion in the problem formulation section of the CDM ERA was based on Aroclors; however, only total PCB concentrations are used in Section 3. There needs to be consistency between the sections. Considering the data available it would be more relevant to discuss all matters in relation to total PCBs rather than Aroclors.

To properly assess the meaning of the data, the detection limit values need to be provided (CDM ERA, Table 4-1). The limits of detection should be specified for all media, particularly for floodplain sediments and soils where no reference data is provided. There are also several other deficiencies in the data used for the whole ERA as illustrated by Table 4-1. First, it is not indicated whether the surface water measurements were performed on filtered water samples or on water containing particulate matter. Further examination of the raw data indicates that the later is the case. The inclusion of particulate matter in these water samples has profound implications for the later uses of this data.

It can be generally observed for the data set that PCB concentrations in biota from reference locations are relatively great (CDM ERA, Table 4-5). For example, a mink at ABSA-1 was estimated to have a whole body PCB concentration of 6.4 mg/kg. It should be noted later that these reference concentrations need to be taken into account when discussing quality criteria and especially when considering remediation criteria.

It is indicated in the text (CDM ERA, page 4-2) that statistical estimates (mean or upper 95% confidence interval) are the most appropriate measures for assessing exposure. While CDM

developed the biota-sampling plan as well as approved all sample sizes, they cite 'limited' data as the reason for choosing the single greatest measured concentration for mice, earthworms, and muskrats as a "...reasonable worst-case exposure potential". The data for PCB concentrations in mice and muskrats are sufficient to calculate meaningful U95 values for many of the study areas within the site. Most contaminant data exhibit a normal or log normal distribution (data distribution was not assessed or discussed in the CDM document). This means that the greatest observed values can be up to several orders of magnitude greater than the majority of values. Therefore, the use of this single point measure will lead to a gross overestimation of exposure concentrations and therefore an unrealistic estimation of contaminant risks. This is particularly true when there are errors in the data values used as there were in the CDM analysis. The use of central tendencies results in a lesser error in these cases. The US-EPA guidance explicitly recommends that a measure of central tendency be used rather than a maximum concentration (US-EPA, 1998). Thus, the central tendency of PCB concentrations should be evaluated together with the U95. The only exception to this is the recommended use of a maximum concentration for the *screening level* assessment at superfund sites (US EPA 1997). To be conservative, the 95% upper confidence of the geometric mean is often used. Use of such an approach would result in a significant reduction in exposure and risk estimates (refer to discussion on uncertainty).

Section 4.1.3 Exposure Scenarios

Aquatic Exposure

The relative risks from exposure to water and sediment containing PCBs will depend on the relative concentrations in those media. It is likely that sediment ingestion/contact will be a considerably more significant exposure pathway than exposure to surface water, particularly if water borne PCBs are bound to particulate matter. Throughout the CDM ERA document, there is a continuing focus on the accumulation of PCBs from surface water. Given the presence of particulate matter in the water samples, it is not possible to model accumulation of PCBs from direct water exposure since these models are based on freely dissolved PCBs and in this case we have no estimate of the freely dissolved PCB concentration.

Terrestrial Exposure

The uptake of PCBs by plants has been shown to be limited particularly from soils (Erickson 1996) and appears to occur only by volatilization of PCBs from soils followed by absorption into the waxy plant cuticle. Current evidence suggests that translocation of PCBs within plant tissues is minimal (Ye *et al.*, 1992; Hill & Napolitano, 1997). This mechanism limits the bioaccumulation of PCBs in terrestrial plants; therefore, the suggested BCF for soil/plant uptake (1.3) is unjustifiably great. Other data suggests that BCFs of 0.01 - 0.1 are more appropriate for estimating plant concentrations of PCBs (Erickson, 1997). Furthermore, the types of PCBs expected to volatilize from soil and be absorbed by plants are most likely lesser chlorinated (*e.g.* mono- and di-chlorinated biphenyls) and thus exhibt less toxicity than the more toxic non-*ortho* substituted PCBs.

Reduction of the soil/plant BCF by a factor 10 to 100 greatly reduces exposure estimates for animals reportedly consuming significant amounts of vegetation such as the robin or red fox (refer to discussion on uncertainty). Dietary intakes of PCBs were estimated to be >90% and 70% derived from plants for robin and fox respectively. It should also be noted that the exposure estimates for robins would be further lessened if their explicit behavior of feeding on plant fruiting bodies had been considered. Plant fruiting bodies are ephemeral and therefore do not receive loadings of PCBs via atmospheric accumulation or translocation within the plant.

Section 4.1.5 Food Web/Food Chain Modeling

In the modeling portion of the ERA, the previously gathered information is used to make exposure estimates for key receptors. In the CDM ERA, estimates of several critical parameters are set to "worst case scenario" values. Therefore, exposure concentrations are based on either maximum observed or 95% UCL concentrations (see discussion above), which is not valid from a statistical or regulatory standpoint. The data for PCB concentrations in mice and muskrats are sufficient to calculate meaningful U95 values for many of the study areas within the site. The central tendency of PCB concentrations should be evaluated together with the U95.

The CDM ERA uses a site foraging frequency (SFF) of 1 for all species. This choice also leads to an overestimation of hazard, as some species will not derive all food intake from the site. This is the case for top predators whose home ranges can be expected to be greater in area than the areas of concern or not conform to the geographic distribution of PCBs. As stated in the ERA, determining home ranges for key receptors can be difficult, but this cannot by itself justify the over-simplified selection of an SFF value of 1. The estimation of NOAEC from LOAEC divided by 10 is a similar over-simplification. The US EPA considers the use of 10 as conservative and values ranging between 1 and 10 should be used when additional information is available. Sufficient published data is available to make better estimates of these values.

In the CDM ERA, the BAF that CDM used is actually a BCF approach. The calculation of fish/water BAFs (CDM ERA, Table 4-6) from these water measurements is an error since the concentrations measured were 'total' concentrations including PCBs bound to particulate matter. BCFs should only be calculated for truly dissolved phase chemicals. If this approach is to be used then some estimate of free/bound PCBs is required to estimate dissolved concentrations followed by recalculation of the BCF values or literature BCF values could be applied.

Similarly, the BSAF approach that CDM used is inappropriate. A BSAF is a relationship between concentrations of a chemical in fish (lipid-normalized) and concentrations in sediment (organic carbon normalized). CDM used wet weight-based fish concentrations and sediment concentrations that were not normalized to organic carbon. This is clearly inappropriate and has implications for the use of BSAFs throughout the remainder of the ERA document.

On p. 4-22, CDM states that for the American robin, measured BAFs were used for earthworms. However, it is not apparent in the exposure calculations that the soil-to-earthworm BAFs were utilized at all. In other words, the BAFs were not utilized to calculate an impoundment-wide earthworm concentration. This type of analysis should have been presented and compared to the

measured earthworm concentrations to indicate the relative sensitivity of the exposure calculation approach.

Section 4.1.6 Uncertainty evaluation – exposure assessment

This section of the CDM ERA does not address issues of uncertainty. Instead, the uncertainty section is used to justify the selection of conservative model parameters. It is also erroneously stated that using "maximum values for the smaller biological data sets, however, is expected to limit uncertainty." In actuality, the choice of a single point estimator for exposure concentrations will greatly increase uncertainty in estimates of risk, particularly since the point chosen is at the extreme end of the frequency distribution.

This section should more appropriately discuss the magnitude of variation in the chosen parameters and how those variations are expected to contribute to the overall estimation of risk. Also required would be a discussion of how variation in the different parameters could be expected to interact (i.e. are variations additive, canceling, or multiplicative).

CDM did not thoroughly evaluate the sensitivity of their risk analyses. Basic sensitivity analyses should have been conducted regarding:

- the PCB content in terrestrial plants. As discussed in previous sections of this report, the concentrations of PCBs that CDM is estimating for terrestrial plants (up to 39.3 mg/kg) is highly questionable and is, in fact, greater than the most contaminated fish (36 mg/kg). When a single estimated parameter has such a major influence on the predicted exposure and risk, it should be measured for the purposes of a baseline risk assessment. To understand the impact of this parameter on the calculated risk, a different value could be used for the estimated plant concentrations. For example, if one assumes negligible PCB concentrations in edible portions of terrestrial plants, then the predicted exposures and risks significantly decrease for the red fox, american robin, mink, and mice. Furthermore, using this

assumption of negligible PCB concentrations in plants, the only receptor that would have a LOAEC-based hazard quotient greater than 1.0 is the mink with a hazard quotient of 5.1.

- the use of a geometric mean rather than an arithmetic mean on p. 4-2, CDM defends their use of the arithmetic mean; however, the log normal distribution of most of the available exposure data indicates that the geometric mean and 95% UCL of the geometric mean are the most appropriate statistics for representing most of the available data. For most parameters, this results in an overestimate of 3 to 20-fold.

- the use of a SFF = 1.0, which is overly conservative. On pp. 4-20 and 4-26, CDM states that, "this ERA does not adjust the SFF and retains the SFF at 1.0...There is no reason to believe ... that predators such as mink will leave the site and obtain food beyond site boundaries." On p. 4-28, CDM states that actual PCB doses probably vary seasonally and spatially. Importantly, however, CDM did not attempt to distinguish between the fraction of diet originating from the Kalamazoo River versus the fraction of diet from adjacent foraging areas (uncontaminated fields, ponds, tributaries, *etc.*) for top predators such as bald eagle and mink. Additionally, CDM did not attempt to account for the fraction of the year that migratory receptor(s) would be present in the Kalamazoo River area. A lack of consideration of both uncontaminated foraging areas within the Kalamazoo River corridor and migratory behaviors leads to an over-estimation of exposure and risk

Section 4.2 Ecological Effects Assessment

While site-specific bioconcentration and bioaccumulation factors should be used in preference to literature values (CDM ERA, Section 4.2), it is important that these values be appropriately derived. As discussed previously, some of the factors derived from site-collected samples are subject to considerable methodological error (e.g., no distinction between dissolved phase and particulate-bound concentrations of PCBs in water). In some of these instances, it would be better to use literature-derived values. The error in these factors is again compounded by selection of the maximum measured values for further analysis.

On p. 4-29, CDM states that, "LOAECs are associated with adverse effects; therefore, PCB concentrations at or near the relevant LOAECs are associated with some risk." The accuracy of this statement depends on the selection of LOAECs from the literature, which are often associated with great uncertainty. Utilizing the LOAEC values that CDM used in their risk assessment, only the mink and American robin have hazard quotients above 1.0, all of the other receptors have hazard quotients below 1.0 at the LOAEC level. Thus, given the definition of CDM, the only receptors that are associated with some risk are the mink and American robin. Given the conservative exposure assumptions (refer to discussion of uncertainty) and questionable toxicity reference doses (refer to the following discussion), it appears that the hazard quotients for both mink and American robin are significantly less than predicted in CDM's ERA.

For the red fox NOAEC and LOAEC, the reference that CDM cites is a book chapter (Grant, 1983) that does not contain a reference to the studies that are discussed and does not provide any toxicological data. From this book chapter, it is mentioned that 2 of the 3 studies with beagle dogs had no adverse effects at a daily dietary concentration of 2.5 mg/kg/d for PCBs (Grant, 1983). Another reference from which a toxicity reference dose can be obtained is an FDA study (Earl *et al.*, 1974) that found a NOAEC and LOAEC of 1 and 5 mg/kg/d for PCBs (Aroclor 1254), respectively. This reference specifically mentions the types of reproductive and developmental effects observed with comparison to controls. Thus, the NOAEC is most likely in the range of 1 to 2.5 mg/kg/d for PCBs in beagle dogs and this should serve as a suitable threshold value for the red fox. Using such a reference dose, the recalculated HQ for red fox is less than 1.0.

There were several inaccuracies in the discussion on mink NOAEC and LOAEC studies from the literature (p. 4-33):

- The reference to Ringer (1983) is not a study, but a book chapter and thus not a primary source of information. Furthermore, it was incorrectly stated that Ringer (1983) fed mink a diet containing 0.64 mg/kg and observed severe reproductive effects in mink. This value of 0.64 is from a study by Platonow and Karstad (1973) which is generally considered "inappropriate for criteria development in part because of possible contamination of feed by other contaminants was not investigated" (GLWQI, 1995). In this same reference, Ringer (1983) presents data from Aulerich and Ringer (1977) to show a clear dietary NOAEC of 2 mg/kg for Aroclors 1016, 1221, and 1242, and a dietary LOAEC of 2 mg/kg for Aroclor 1254.
- The reference to the GLWQI is also not a primary study. The GLWQI recommends the use of 2 mg/kg as a dietary LOAEC from the Aulerich and Ringer (1977) study discussed above.
- The Heaton et al., (1992) study identified a LOAEC from a mink feeding study using fish from Saginaw Bay. It is unknown how comparable the PCB congener distributions or how comparable would be the concentrations of co-contaminants (especially polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans) are between Saginaw Bay and the Kalamazoo River.
- The Aulerich et al., (1985) study that is cited did not determine a dietary LOAEC for mink of 0.25 mg/kg; rather the dietary LOAEC from this study is clearly stated as 2.5 mg/kg.

Finally, all of the NOAECs described above were derived by dividing the LOAEC by 10 even though CDM states on p. 4-29 that, "this ERA ... considers LOAEC/3 where NOAECs are estimated for mink with dietary exposures." Why then were the LOAECs divided by 10 rather than 3? These inaccuracies (including ones previously discussed) all erred on the side of conservatism and contribute to an overestimation of risk by at least 3 to 10-fold. Furthermore, since the LOAECs and NOAECs for mink are utilized to calculate the effects thresholds in fish as prey (0.22 mg/kg), surface water (0.00038 µg/L), and sediments (0.12 mg/kg), these errors carry through to the risk characterization and remediation discussions later in CDM's ERA. Thus, the errors committed in deriving a value for mink have significant impact on and raise concerns regarding the validity of the remaining calculations based on this value.

4.2.3 Uncertainty Evaluation- effects assessment

This section should include some form of sensitivity analysis to determine the effects of changing critical parameters, particularly as such conservative estimates were used. See comments on Section 4.1.6 above.

Summary of Section 4

Throughout the ERA process CDM has chosen to use the most conservative estimates of critical parameters. These include

- Greatest observed concentration data for mice, earthworms, and muskrats
- Site Foraging Frequency (SFF) value of 1.0
- Inconsistency in using a geometric mean approach for effects data and an arithmetic mean approach for exposure data
- Unreasonably great soil/ plant BCF
- NOAEC = LOAEC/10

Together these estimates, while conservative, introduce an unacceptable level of bias and uncertainty into the ERA.

Section 5 Risk Characterization

As mentioned in previous sections there are several significant deficiencies in the section. The comparison of water concentrations of PCBs with the "KRSS-specific effects concentration" value is particularly not relevant. The reference sites included in Figure 5-1 are more than an order of magnitude above this 'effects' concentration. This places considerable question on the relevance of these KRSS-specific 'effects' concentrations. It would also be interesting to compare this value to the method detection limits to see whether non-detect values were greater than these 'effects' concentrations.

'Effects' concentrations used in the assessment of streambed sediments are also of concern. The 'effects' concentrations indicated in Figure 5.2 are all based on either the same erroneous water PCB concentrations or the questionable fish tissue limit of 0.22 mg/kg discussed previously. In view of the deficiencies of the water measurements and fish tissue limit calculations, the 'effects' concentrations presented in Figure 5.2 need to be reevaluated.

It must be indicated that the KRSS derived 'effects' concentrations are not actually concentrations associated with effects as indicated in section 4.2.1 and Table 4-7. The 'effects' concentrations derived from KRSS data are only crude estimates of no effect levels. No attempt has been made to measure any adverse effects in KRSS wildlife. It would therefore seem appropriate to include some of other 'effects' concentrations presented in Table 4.7 (and an estimate of confidence intervals derived from a sensitivity analysis) on Figures 5.1 to 5.4 rather than rely solely on the derived KRSS 'effects' concentrations.

Table 5-1 illustrates the deficiencies in using a simple extrapolation of LOAEC to NOAEC by using a factor of 10. It is difficult to justify using the same scaling factor for a LOAEC based on 15% mortality in rats and a LOAEC based on reduced sperm production in American Kestrel. Toxicological data need to be relatively comparable to ensure that the relative hazard to each species can be compared as indicated in Table 5-3.

In Table 5-2, the derived PCB concentrations for plants are too great compared to other food items and to literature values (Erickson, 1997). This results in vegetation being estimated as the major dietary source of PCBs in several species most notably the red fox, American robin and white-footed deer mouse. These great estimates of PCBs in plants result from the use of a plant BAF of 1.3. This error requires that the ERA procedure for terrestrial species be revised.

Table 5-3, presents the essential results of the CDM analysis by presenting ranked relative risk to each receptor. No indication is provided of the criteria used to rank the different receptors. In addition, as this is a hazard assessment, it is essential that the HQ ranges for each receptor be

provided. For example, HQ ranges could be presented as part of a sensitivity analysis. Also, a set of HQs based on NOAEL and LOAEL TRVs, mean, and U95 exposures would be informative.

While not mentioned in section 4 of the CDM ERA, the bald eagle appears in Table 5-3. Since bald eagles nest in the Allegan State Game Area, it is appropriate to correct the "transient and accidental" characterization in Appendix A. In the case of the bald eagle, an SFF of 1.0 appears to be inappropriate (CDM ERA, Appendix C).

The discussion in section 5.1.4 concerning the abundance of carp and their significance as a food source to mink is questionable. Carp tend to frequent deeper water and mink therefore only have ready access to them during the fish's breeding season. It is also questionable whether mink can effectively predate fish considerably larger than themselves. Similarly, carp constitute only a small proportion of the diet of bald eagles in Michigan (Bowerman 1993).

Section 5.3 Risk Summary and Ecological Significance

Given the bias introduced by the use of a 'worst-case scenario', the word "likely" is out of place in section 5.3. The presentation of HQ ranges for each species is also required, as is an unambiguous indication of what the HQ value means (see introduction to this document). Without this, the uninitiated reader of Table 4-3 could assume all indicated species were in danger from PCB contamination.

On page 5-27, secondary data on the sampling of mink is provided. If this data is relevant then it should be used in the assessment sections 4 and 5. However, as the data is of questionable value, it should be omitted. No association between mink populations and PCB contamination can be derived from this data due to its limited extent and questionable sampling protocol.

Section 5.4 Uncertainty Evaluation – Risk Characterization

The uncertainty evaluation section of section 5 of the CDM ERA is again inadequate. The CDM ERA simply restates the previously discussed uncertainties but does not attempt to assess their impact on the results of the assessment. A more useful approach would be to conduct a 'sensitivity analysis' in which values for the various uncertain parameters are altered and the result on the assessment is discussed. Given the great levels of uncertainty in the CDM ERA, this approach is particularly important as it provides information for further refining the assessment and in identifying those parameters that most contribute to uncertainty. This exercise would also identify data gaps, which can provide direction for future sampling and analytical efforts.

The final paragraph of this section (page 5-28) is particularly worrying. We find the evidence provided in this assessment less than 'overwhelming' and that none of the previously stated 'hypotheses' have been proved. This statement suggests some form of statistical rigor to the assessment that simply does not exist.

Section 4.5 Remediation Issues

This baseline ERA, which is more accurately termed as a screening level risk assessment, is not designed to provide adequate information for remedial action decisions. As such, discussion of PCB remediation in a screening level environmental risk assessment is premature. Statements like "PCB remediation will result in removal of risks from other chemical stressors" are not based on data and thus are unjustifiable. The discussion of site specific and media specific cleanup goals are not only premature but also reckless. Here, the proposed media-specific cleanup goals are based on erroneously derived exposure estimates resulting from the numerous deficiencies and miscalculations discussed above.

Section 5 References and Appendices

The citation of Mehne (1994) is incomplete; specify report title or "personal communication".

The habitat and feeding characteristics for the bald eagle should be presented in Appendix B.

Summary

The major weakness of the CDM ERA is the use of parameters based on the "worst-case scenario". The application of very conservative assumptions and use of biased parameter estimates yields a gross overestimate of potential risks presented by PCBs to organisms at the KRSS.

The errors encountered in BCF and BAF calculations make the results of this assessment unusable and further increase the risk estimates already elevated by the use of very conservative assumptions for environmental concentrations, SFF, toxicity reference values, and other parameters.

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Subject:
Camp Dresser & McKee Kalamazoo River Baseline Ecological Risk Assessment

Date:
July 16, 1999

Dear Mark:

The following comments are being submitted by JSA Environmental/ARCADIS Geraghty & Miller to address inadequacies and uncertainties in the *Final Baseline Ecological Risk Assessment for the Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site*, prepared by Camp Dresser & McKee (CDM) for the Michigan Department of Environmental Quality and released in June 1999. The comments address (1) CDM's risk assessment for the mink, the most sensitive ecological receptor at the site, (2) the exposure assessment for other wildlife receptors of interest, (3) the effects assessment for other wildlife receptors of interest, (4) the assessment of direct toxicity to fish and invertebrates, and (5) analytical considerations.

Contact:
Kenneth D. Jenkins

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(562) 628-1176

Most of the problems identified in the CDM assessment create an over-conservative bias. Both the exposure and effects assessment are over-conservative for the aquatic-feeding mink and bald eagle, and PCB exposures for terrestrial receptors are dramatically over-estimated. The effects assessment for the robin, white-footed mouse, and muskrat is under-conservative; however, for the robin and mouse, the under-conservative bias is overshadowed by the extreme over-conservatism of the exposure assessment. The bias in the muskrat assessment is uncertain, but as a plant consumer, this species would clearly be expected to experience lower PCB exposure than other wildlife receptors of interest such as the mink. The remediation goals established in the CDM assessment are driven by alleged risk to mink (sediment and surface water) and to robins (soil). Overall, the comments provided below indicate that the CDM assessment advocates unnecessarily low remediation goals for surface water, sediment, and soil at the Kalamazoo River site.

Should you have any questions or comments, please feel free to call.

Sincerely,

for

Kenneth D. Jenkins, Ph.D.
President

KB13000533

1. MINK ASSESSMENT

CDM's risk assessment for the mink merits particular scrutiny, because the risk calculations for this receptor determine the proposed remediation goals for sediments and surface water at the site. Due to a series of errors and inappropriate assumptions, the remediation goals developed by CDM are over-conservative by at least an order of magnitude, as discussed below.

1.1 Mink diet

CDM's risk assessment for the mink contains four different sets of assumptions regarding dietary composition:

- The food web model used to calculate hazard quotients (Appendix C; Table 5-3) assumes a diet of 38% fish (consisting of equal amounts of carp, suckers, and bass), 28% mammals (equal amounts of muskrats and mice), 13% herptiles (i.e., reptiles and amphibians), 9% birds, 7% plants (equal amounts of terrestrial and aquatic plants), and 5% aquatic invertebrates. This dietary composition is based on U.S. EPA guidance.
- The proposed remediation goals for surface water and sediment assume a diet of 100% carp.
- Section 5.1.4 qualitatively describes a diet consisting primarily of carp, supplemented with mice, muskrats, and crayfish.
- Appendix B describes a diet containing 30% fish, 30% mammals, 20% herptiles, and 20% birds, based on CDM (1994) (which is not included in the reference section).

It is clear that the development of remediation goals based on carp as the sole food item for mink is not representative of the feeding habits of mink. It is an over-simplification for mathematical convenience that results in unnecessarily low remediation goals, since carp contain the highest PCB levels of any mink food item and, in the worst case presented above in the food web model, comprise 38% of the minks diet (this also assumes that carp represent 100% of fish consumed).

The first step in identifying appropriate risk-based concentrations for the mink is to reasonably estimate the composition of the mink's diet. For each food item, site-specific or literature-based information is then used to relate PCB concentrations in prey to PCB concentrations in water, sediment, or soil. Risk-based concentrations in environmental media can be back calculated using the food web model.

A simple example illustrates the potential effect of dietary assumptions on calculated remediation goals. The average (arithmetic mean) biota-sediment accumulation factor (BSAF) for carp is 1.9, whereas the arithmetic mean BSAF for all fish is 1.0 (see Table 4-6). Thus, the assumption that mink consume all types of fish instead of 100% carp would increase the remediation goal for sediment by approximately a factor of 2. Adding benthic invertebrates (e.g. crayfish) to the diet would be expected to further increase the remediation goal, because invertebrates contain less lipid in their tissue than fish and accumulate somewhat lower PCB levels on a wet weight basis.

1.2 Mink Toxicity Reference Values (TRVs)

Two aspects of the TRVs developed for the mink in Section 4.2.1 are inappropriate. First, the Aulerich et al. (1985) lowest observed adverse effect concentration (LOAEC) citation is incorrect. Aulerich et al. (1985) observed reproductive failure in mink fed 2.5 mg PCB/kg fresh weight diet. CDM incorrectly cites this study's LOAEC as 0.25 mg/kg instead of 2.5 mg/kg. This error significantly decreases the estimated LOAEC, calculated as the geometric mean of four observed LOAECs.

Second, it is unnecessary to estimate no observed adverse effect concentrations (NOAECs) from LOAECs, because NOAECs can be directly identified from the literature. The relationship between a NOAEC and a LOAEC is determined purely by the selection of test concentrations; an actual threshold level lies somewhere between the NOAEC and LOAEC. Thus, experimentally observed NOAECs are always preferable to arbitrarily estimating NOAECs from LOAECs.

We are aware of three experimentally observed NOAECs for PCB effects on mink reproduction. Aulerich and Ringer (1977) provide a NOAEC and LOAEC of 1 and 2 mg PCB/kg fresh weight diet, respectively. This study provides the basis for the Great Lakes Initiative LOAEC cited by CDM. Aulerich and Ringer (1977) evaluated mink reproduction based on number of kits and kit survival, but not kit growth. Wren et al. (1987) observed no effect on these endpoints for mink exposed to 1 mg PCB/kg fresh weight diet, similar to Aulerich and Ringer (1977), but kit growth was significantly reduced at this exposure level. The authors infer that this effect (30% reduction in offspring body weight) would probably be linked to poor survival and reproductive potential of the offspring in the wild (Wren et al., 1987). A NOAEC based the most sensitive test endpoint (kit growth) is available from Hornshaw et al. (1983), who fed mink Great Lakes fish containing PCBs. Hornshaw et al. (1983) observed no significant effects on mink reproduction at exposures of 0.21, 0.48, and 0.63 mg PCB/kg fresh weight diet, while adverse effects were observed at 0.66, 0.69, and 1.5 mg PCB/kg fresh weight diet. The NOAEC observed by Wren et al. (1987) of 0.5 mg/PCB/kg fresh weight diet is consistent with the findings of Hornshaw et al. (1983).

Based on the studies cited here and in the CDM report, a LOAEC for the mink can be recalculated as the geometric mean of 0.64 (Ringer, 1983), 2 (Great Lakes Initiative/Aulerich and Ringer, 1977), 0.72 (Heaton, 1992), 2.5 (Aulerich et al., 1985), 1 (Wren et al., 1987), and 0.66 (Hornshaw et al., 1983). The resulting LOAEC is equal to 1.1 mg PCB/kg fresh weight diet. Based on a NOAEC of 0.63 mg PCB/kg fresh weight diet from Hornshaw et al. (1983), a maximum acceptable test concentration (MATC) for the mink is recalculated as 0.8 mg PCB/kg fresh weight diet. For comparison, CDM calculated a MATC of 0.22 mg PCB/kg fresh weight diet.

It is important to note that the results of Hornshaw et al. (1983) are potentially biased low with respect to PCBs alone, because the PCB-contaminated fish used in this study may also have contained other toxic chemicals. This uncertainty also applies to the Heaton (1992) LOAEC cited by CDM. Thus, the LOAEC and NOAEC values identified above should be considered conservative. The calculation of a MATC as the geometric mean of a LOAEC and NOAEC parallels the procedures followed by CDM; however, it should

be noted that this is essentially an arbitrary calculation used to facilitate risk management and is technically less defensible than the identification of NOAEC and LOAEC values.

An additional consideration is the uncertainty associated with extrapolating the results of laboratory experiments to mink populations in the field. Specifically, the level of effect required to produce a measurable, population-level effect is uncertain. For instance, a level of PCB exposure which causes reduced kit growth without reduced kit survival in the laboratory may cause reduced kit survival and mink population decline in the field (as postulated by Wren et al., 1987), or it may simply result in a sustainable resident population of small mink. Also, factors other than PCB exposures (such as physical habitat modifications and exposures to other chemicals) may limit the reproductive potential of mink populations in parts of the Kalamazoo River site.

1.3 Bioaccumulation Factor (BAF) and BSAF Calculation

CDM calculates an average LOAEC TRV as a geometric mean (assumes a lognormal distribution), whereas average BAF and BSAF values are calculated as arithmetic means (assumes a normal distribution). In general, the geometric mean tends to be lower than the arithmetic mean for a given data set. Thus, CDM's arbitrary selection of different mean calculation methods for different aspects of the assessment tends to increase the estimated risk.

Although not used by CDM, statistical tests are available to determine whether data are distributed normally or lognormally, given a sufficiently large sample size (n>8). A lognormal distribution is typically assumed for small data sets, because environmental data tend to be distributed lognormally. For each fish species, 7 BSAFs (sediment-based) and 4 BAFs (surface water-based) are provided by CDM (Table 4-6). Using geometric instead of arithmetic means, the average BSAF for all fish is 0.56 instead of 1.0.

1.4 Cumulative Effect on Remediation Goals

The cumulative effect of correcting the errors described above is estimated as a 10- to 15-fold increase in the remediation goals developed for sediments and surface water to protect mink from exposures via the food web. Other factors may lead to further increases in the remediation goals (for example, see discussion of exposure statistics, below).

2. EXPOSURE ASSESSMENT FOR WILDLIFE

2.1 Exposure Statistics

In general, CDM estimates exposures to PCBs in various environmental media as the arithmetic mean and the arithmetic 95% upper confidence limit of the mean (U95) (Table 4-6; Appendix C). These calculations assume the PCB data are normally distributed, whereas environmental data are typically lognormally distributed. When enough data are available (n>8), a simple statistical test to identify the distribution type should be conducted to support which type of mean and U95 to use.

Additionally, hazard quotients are calculated using U95 exposure values (or maximum values; see below) but not mean values. Mean exposure concentrations are evaluated only graphically and only for a subset of the receptors of interest. Once errors in the food web model are corrected, it would be more useful to calculate hazard quotients using both mean and U95 exposures. The U95 is designed to overestimate the true mean PCB concentration 95% of the time and is used conservatively to assess the uncertainty associated with using a limited data set. If calculated correctly, the mean provides the most likely exposure concentration for wildlife in a given study area. Together, the mean and U95 indicate the difference between the most likely and reasonable maximum exposure levels.

2.2 Use of Maximum PCB Concentrations

CDM uses the U95 to estimate exposures to PCBs in fish tissue, surface water, and sediments. However, maximum concentrations are used for mouse, muskrat, and earthworm tissue exposures (Table 4-5; Appendix C). This is inappropriate, as the amount of data available for terrestrial prey items is similar to that available for fish. For each ABSA, 11 fish tissue samples were analyzed for each species, compared to 10 mouse tissue samples per TBSA. Depending on the sample location, either 6 or 12 muskrat tissue samples were analyzed. Tissue data are limited for earthworms at most locations (n=3), except TBSA 5 (n=24).

The mathematical procedure used to calculate the U95 takes into account the number of samples being evaluated, as well as the variability of the data set. If the number of samples is too small to provide a conservative characterization of the central tendency of the data, the U95 will exceed the maximum measured concentration. Therefore, it is appropriate to use the U95 in all cases except when the U95 exceeds the maximum measured concentration. An alternative approach for evaluating the earthworm data is to develop a site-specific soil-earthworm accumulation factor for application to the U95 for soil. Either of these approaches is more appropriate than the over-conservative use of maximum PCB concentrations for terrestrial prey species.

2.3 Terrestrial Plant Exposures

Exposures of terrestrial-feeding wildlife to PCBs are drastically over-estimated due to the use of an unrealistically high soil-plant bioaccumulation factor. CDM acknowledges that terrestrial plant uptake of PCBs is over-estimated and ultimately dismisses the finding of risk to terrestrial receptors (pages 4-25, 5-24). Thus, the terrestrial risk assessment provides essentially no useful information to risk managers. A useful assessment would either (1) develop a reasonable soil-plant bioaccumulation factor from the published literature, or (2) use site-specific measurements of PCB bioaccumulation in plant tissue.

Despite its dismissal of the risk calculations for terrestrial receptors, CDM uses the same calculations (specifically for the robin) to develop a remediation goal for soil. The similarity between the remediation goals for soil (robin) and sediment (mink) is used as a justification for the soil remediation goal (page 5-31). This is a meaningless observation, as there is no inherent mechanism that would suggest a parity between these two calculations. In fact, an appropriate remediation goal for soil would probably be much higher than

for sediment, because the potential for bioaccumulation is much less in terrestrial systems than in aquatic systems.

A review of soil-plant accumulation factors for 29 organic chemicals (Travis and Arms, 1988) provides an indication of the extent to which CDM has overestimated PCB exposures via plant consumption. Based on a quantitative relationship between bioaccumulation and K_{ow} and on data collected specifically for Aroclor 1254, an appropriate soil-plant accumulation factor is approximately 0.02. This is 65 times lower than the soil-plant accumulation factor of 1.3 used by CDM.

Interestingly, one rationale provided for using an admittedly over-conservative soil-plant bioaccumulation factor is that wetland plants might accumulate PCBs to a higher degree than terrestrial plants, such that the assessment would be less over-conservative in wetland or flooded portions of the site (page 4-25). However, this rationale is inconsistent with the CDM food web model for the muskrat, which eats emergent aquatic vegetation such as cattails. The food web model predicts PCB concentrations in aquatic plants not using the soil-plant bioaccumulation factor, but rather by applying a water-plant bioaccumulation factor to surface water concentrations (Appendix C). As a result, PCB concentrations are estimated to be much lower in wetland plants than in terrestrial plants, in direct contradiction to the rationale for the terrestrial exposure assessment methods. Neither of CDM's methods of estimating PCB concentrations in plants is appropriate, since the water-plant bioaccumulation factor used for aquatic plants is derived from a study using marine diatoms, which may show different bioaccumulation characteristics than freshwater macrophytes.

2.4 Site Foraging Frequency

CDM assumes that all wildlife receptors of interest obtain 100% of their diet within the Kalamazoo River site (page 4-26). This assumption is probably over-conservative for larger, terrestrial species such as the red fox and great horned owl. For aquatic-feeding species, the site is large enough to support wildlife populations; however, uncontaminated tributaries and other areas outside the site could provide some part of the diet of resident mink and bald eagles.

3. EFFECTS ASSESSMENT FOR WILDLIFE

3.1 TRV Calculation

In developing TRVs for wildlife receptors of interest, CDM has incorrectly calculated test species doses from test species exposure concentrations (Appendix C). Frequently, toxicity studies present the PCB concentration in the diet of test organisms (mg/kg) rather than the dose (mg/kg body weight/day). The application of test data across species must be conducted on the basis of dose rather than exposure concentration. Dose levels are calculated for laboratory tests based on body weight and ingestion rate information for the laboratory test species. For mammals, the test species dose may be modified for a receptor of interest using a body weight extrapolation factor (see below); however, this type of extrapolation is not valid for birds. CDM incorrectly used body weight and ingestion rate information for the receptors of interest instead of the laboratory test species to calculate test species doses from dietary concentrations.

This error will tend to result in inflated TRVs for small receptors of interest (e.g., robin TRV of 6 mg/kg/day rather than 1.8 mg/kg/day; see below) and over-conservative TRVs for larger receptors of interest (e.g., eagle, owl). The only receptor of interest which is not affected by this error is the mink, since interspecies extrapolation is not required for this species. For small species, the increased conservatism of correctly calculated TRVs will be more than offset by correctly estimating PCB exposures in plants and other terrestrial food items (see above).

3.2 Benchmark Tests for *Bird* TRVs

Two sets of TRVs are provided in the CDM risk assessment for birds. All are incorrect, as stated above. Additionally, Appendix C and the "recommended dietary threshold" column of Table 5-1 use an observed NOAEL from McLane and Hughes (1980), a screech owl study. The "NOAEC" column of Table 5-1 and Table 5-3 use one-tenth of the LOAEL instead of an observed NOAEL. Observed NOAELs are preferable to estimated values, as stated above.

Additionally, the LOAEL for the robin is based on a study using chickens (Appendix C). Where possible, TRVs should be derived using wildlife species rather than domesticated species. Chickens are genetically altered through breeding, are less ecologically relevant than wildlife species, and are more sensitive to PCBs than wildlife species. The Great Lakes Initiative uses a pheasant study (Dahlgren et al., 1972), which provides a LOAEL of 1.8 mg/kg/day. NOAEL values for wildlife species are available from the screech owl study cited above (0.3 mg/kg/day), and a mallard study (Custer and Heinz, 1980) which provides a NOAEL of 1.5 mg/kg/day.

3.3 Mammal TRVs

The TRVs identified for the mouse, muskrat, and red fox in Appendix C do not reflect either an up-to-date review of toxicity studies or an understanding of interspecies extrapolations for mammals. The NOAEL TRVs for the mouse and muskrat (0.5 mg/kg/day) are similar to other available NOAEL values, but the LOAELs for these receptors are notably non-conservative. For the mouse, the LOAEL TRV of 6.5 mg/kg/day (Sanders and Kirkpatrick, 1977) is based on a behavioral effect that is not directly relevant to population-level impacts, and is higher than other LOAELs from mouse studies, including:

- 1.5 mg/kg/day from a two-generation study of reproductive success in white-footed mice (Linsey, 1988), and
- 1.27 mg/kg/day from a three-generation study of reproduction and developmental effects in oldfield mice (McCoy et al., 1995).

For the muskrat, CDM's LOAEL of 150 mg/kg/day (Kimbrough et al., 1972) represents a concentration associated with 15% mortality in rats. This is much less conservative than other rat LOAELs such as:

- 1.5 mg/kg/day for decreased litter size following 129 days exposure (Linder et al., 1974), and
- 1.3 mg/kg/day for a 42-day study of neonatal mortality (Overman et al., 1987).

Because many studies have evaluated the effects of PCBs on rats and mice, it may be appropriate to develop NOAEL and LOAEL TRVs based on multiple studies, similar to the procedure used by CDM to develop TRVs for the mink. Since the NOAEL TRVs for the mouse and muskrat are unlikely to change greatly, the NOAEL-based hazard quotients for these receptors would be most affected by corrections in the exposure assessment rather than the effects assessment. LOAEL-based hazard quotients currently are not calculated in the CDM report but would be useful following recalculation of appropriate LOAEL TRVs.

For the red fox, CDM was unable to identify an appropriate LOAEL TRV, while the NOAEL TRV is taken from a study in which no toxicity was observed at any test concentration. CDM has attempted to avoid interspecies extrapolations by identifying toxicity studies which used laboratory species that are closely related to the receptors of interest; however, this approach is unsuccessful for the red fox due to the limited number of toxicity studies conducted using dogs. In this case, it may be preferable to use toxicity data from other species (such as mice and rats) and a body-weight extrapolation factor to develop technically sound TRVs.

The use of body weight scaling factors is based on numerous studies showing that many physiological functions such as metabolic rates and responses to chemicals are a function of body size for mammals (Sample et al., 1996). Body weight scaling factors are calculated according to:

$$NOAEL_w = NOAEL_t \times \left(\frac{BW_t}{BW_w} \right)^{0.25}$$

where:

$NOAEL_w$	=	NOAEL for the mammalian receptor of interest;
$NOAEL_t$	=	NOAEL for the mammalian test species;
BW_t	=	Test species body weight; and
BW_w	=	Wildlife receptor of interest body weight.

As an example, a LOAEL of 1.5 mg/kg/day for laboratory rats (0.35 kg) translates to a TRV of 0.78 mg/kg/day for the red fox (4.7 kg) and a TRV of 1.1 mg/kg/day for the muskrat (1.4 kg). While TRVs for mammals decrease with body weight, exposure also decreases due to decreasing ingestion rates relative to body weight.

It should also be noted that Table 5-1 lists "recommended dietary threshold values" for muskrat, mouse, and fox as the NOAEL TRVs divided by 100. These values are unexplained and do not appear to be used in the risk assessment. Dividing NOAEL TRVs by 100 provides arbitrary values which would not be useful for risk assessment purposes.

3.4 Summary of Cumulative Impact on Wildlife Risk Assessment

In general, the most important factor influencing the risk assessment for wildlife receptors other than the mink is the extreme overestimation of PCB exposures, especially for terrestrial receptors. The only exception is the muskrat, for which exposures may be either over- or under-estimated due to the use of an

inappropriate water-plant bioaccumulation factor. The errors noted in the calculation of TRVs should be corrected in order to provide an accurate assessment; however, their impact is expected to be relatively minor relative to the impact of inappropriate exposure assessment procedures.

4. FISH AND INVERTEBRATES

4.1 "Direct Contact" Assessment

The use of the term "direct contact" in Sections 5.1.1 through 5.1.3 is a misnomer, as the risk characterization described in these sections is based solely on bioaccumulation-related impacts on wildlife. The CDM report does not include a quantitative risk assessment for direct contact (i.e., risks to fish or aquatic, benthic, or terrestrial invertebrates). Even the U.S. EPA ambient water quality criterion for PCBs is based on potential risks to wildlife; a criterion developed solely for the protection of aquatic life would be much higher. Thus, CDM's comparison of surface water, sediment, and soil PCB concentrations to various risk-based concentrations should in no way be construed as indicating the potential for adverse effects on fish or invertebrates. The CDM report is internally inconsistent on this point, stating on page 5-14 that "PCB concentrations in the Kalamazoo River and Portage Creek surface water and streambed sediment clearly pose substantial risks to aquatic biota, including aquatic invertebrates and fish (Table 5-3)," while Table 5-3 indicates a "low toxicity risk" associated with aquatic organisms' exposures to surface water.

The focus on wildlife in CDM's risk assessment is appropriate, because adverse ecological effects due to direct contact with PCBs occur at much higher environmental concentrations than effects related to bioaccumulation (U.S. EPA, 1997). For instance, in a recent review Niimi (1996) found that adverse effects on fish survival, growth, and reproduction correspond to adult fish tissue PCB concentrations above 25 mg/kg, whereas effects on fish consumers are expected at lower fish tissue levels. Similarly, experiments focusing on the toxicity of PCBs in sediment to benthic invertebrates indicate relatively high toxicity thresholds. Murdoch et al. (1997a; 1997b) observed no toxicity in PCB-spiked sediments at the highest concentrations tested (26 mg/kg normalized to 1% total organic carbon[1]). Swartz et al. (1988) identified an LC50 for Aroclor-1254 of 29 mg/kg normalized to 1% total organic carbon, in a spiked sediment test that did not include an equilibration period, thus increasing PCB bioavailability. DiPinto et al. (1993) observed decreased copepod reproduction in sediment spiked with 1 mg/kg normalized to 1% total organic carbon; again, this test may have been over-conservative due to the lack of an equilibration period after sediment spiking. Ho et al. (1997) identified aqueous LC50s for application to PCB concentrations in porewater of 10-100 µg/L; these levels correspond to sediment concentrations of approximately 30-300 mg/kg

[1] Sediment PCB concentrations are normalized to 1% total organic carbon for comparison of effect concentrations among sediments, because organic carbon is the most important factor controlling the bioavailability of hydrophobic organic chemicals to benthic invertebrates. An effect concentration of 10 mg/kg normalized to 1% total organic carbon is equivalent to an effect concentration of 80 mg/kg dry weight for a sediment containing 8% total organic carbon. By comparison, the site-specific sediment remediation goals developed for the protection of wildlife species are not normalized to 1% total organic carbon, because they are based on site-specific bioaccumulation factors which account for the effect of organic carbon on PCB bioavailability.

normalized to 1% total organic carbon. The high organic carbon levels in Kalamazoo River sediments (approximately 8%) are expected to further limit PCB toxicity to benthic invertebrates.

It should be noted that various sediment quality benchmarks have been developed for PCBs using large data sets for sediments contaminated primarily with chemicals other than PCBs. These benchmarks are typically highly over-conservative and are not predictive of toxicity, even within the databases used to derive them (e.g., Long et al., 1995). This results from the use of association-based (as opposed to cause-effect-based) evaluation methods for sediments contaminated with complex chemical mixtures. These methods are not able to distinguish which chemicals are important contributors to toxicity, and they produce highly over-conservative benchmarks for those chemicals which are not important contributors to sediment toxicity (Fuchsman et al., 1998). The available cause-effect-based studies for PCBs (cited above) indicate that direct toxicity thresholds for PCBs in sediment are much higher than the generic benchmarks cited by CDM in Table 4-7. CDM appropriately omits these benchmarks from the risk characterization, acknowledging their extreme uncertainty and lack of applicability to the Kalamazoo River. However, the citation of these benchmarks in Table 4-7 is misleading, particularly the citation of 0.37 mg/kg as a "concentration at which adverse effects are always observed," which is incorrect and does not reflect the conclusions of Long and Morgan (1991).

5. ANALYTICAL CONSIDERATIONS

It is generally recognized that analytical methods that are designed to detect technical-grade Aroclors are not ideally suited for environmental analyses and alternative methods of analyzing PCBs in environmental samples have become available in recent years. However, the effects data available for assessing ecological risks are generally derived from experiments using Aroclors. CDM has sidestepped the Aroclor issue by expressing all PCB concentrations as total PCBs (presumably calculated by summing detected Aroclor concentrations). This approach is limited by (1) the accuracy of the Aroclor analyses which is dependent on the specific analytical techniques employed, and (2) the inability to evaluate differences in toxicity associated with different PCB congeners. CDM fails to acknowledge these sources of uncertainty.

An important consideration in any future evaluations of PCBs at the site will be the comparability of exposure and effect concentrations. For instance, the Hornshaw et al. (1983) study cited above in the discussion of mink TRVs, in which mink were fed contaminated Great Lakes fish, provides effect concentrations which represent (1) the type of congener mixture that tends to accumulate in fish, and (2) the type of Aroclor analysis used for Kalamazoo River fish. The comparability of the resulting effects data for evaluating site-specific exposure concentrations is thus relatively good, although the PCB congener composition and concentrations of other chemicals in the Great Lakes fish may have differed from those occurring in Kalamazoo River fish. If other analytical methods are employed at the site in the future, the effects data would need to be corrected for any bias identified in the Aroclor method compared to the alternative PCB analysis.

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